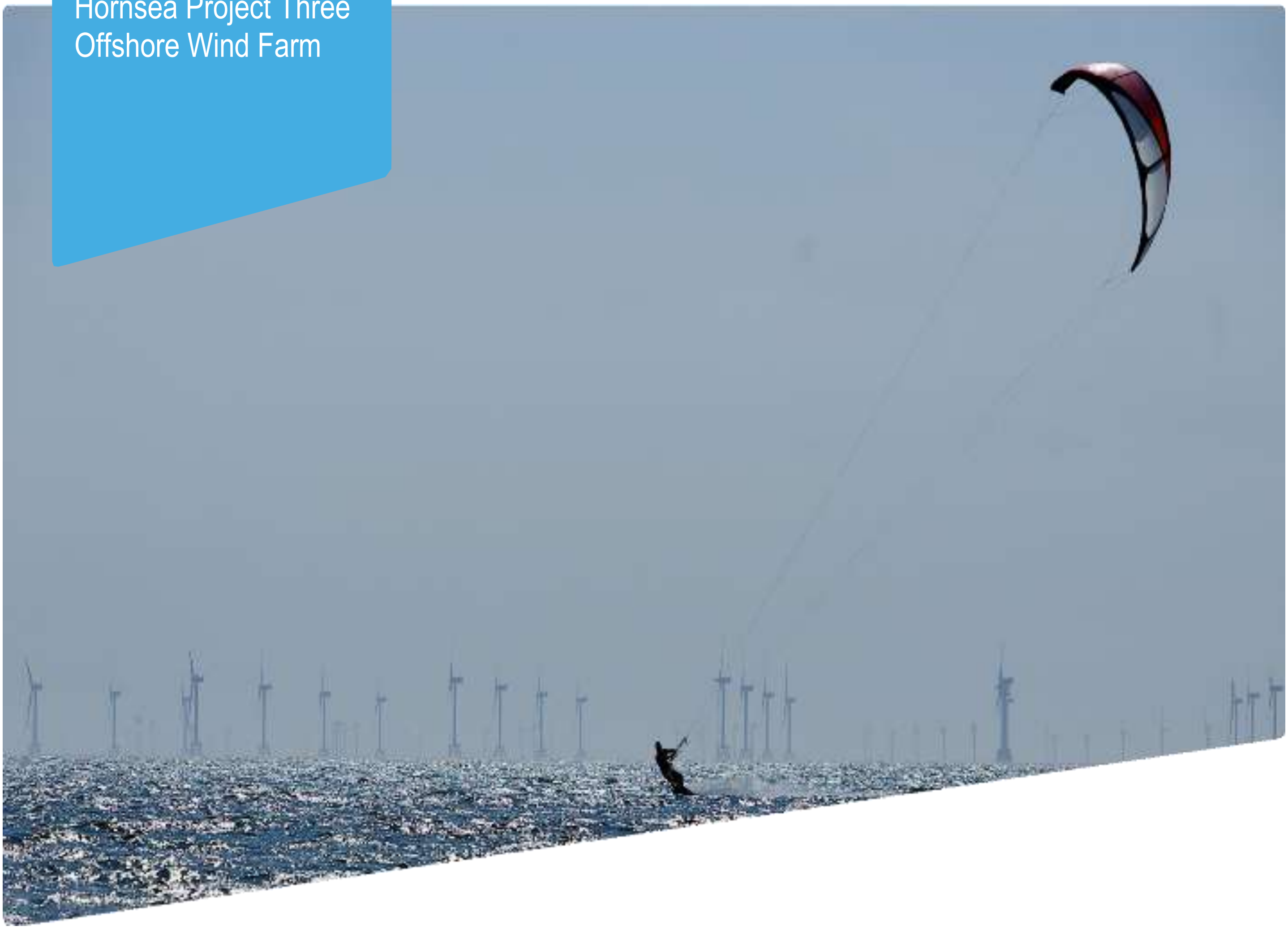
Hornsea Project Three
Offshore Wind Farm



Hornsea Project Three

Offshore Wind Farm

Appendix 11 to Deadline 7 submission – Furness et al., 2013

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Hornsea 3
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Front cover picture: Kite surfer near a UK offshore wind farm © Ørsted Hornsea Project Three (UK) Ltd., 2019.

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Assessing vulnerability of marine bird populations to offshore wind farms

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ABSTRACT

Offshore wind farms may affect bird populations through collision mortality and displacement. Given the pressures to develop offshore wind farms, there is an urgent need to assess population-level impacts on protected marine birds. Here we refine an approach to assess aspects of their ecology that influence population vulnerability to wind farm impacts, also taking into account the conservation importance of each species. Flight height appears to be a key factor influencing collision mortality risk but improved data on flight heights of marine birds are needed. Collision index calculations identify populations of gulls, white-tailed eagles, northern gannets and skuas as of particularly high concern in Scottish waters. Displacement index calculations identify populations of divers and common scoters as most vulnerable to population-level impacts of displacement, but these are likely to be less evident than impacts of collision mortality. The collision and displacement indices developed here for Scottish marine bird populations could be applied to populations elsewhere, and this approach will help in identifying likely impacts of future offshore wind farms on marine birds and prioritising monitoring programmes, at least until data on macro-avoidance rates become available.

1. Introduction

To meet targets for a reduction in greenhouse gas emissions, many Governments, especially within the European Union but also in North America, are encouraging the development of renewable energy generation such as offshore wind farms. There is the potential however for such developments to have adverse effects on the environment and particularly on marine birds through collision mortality, disturbance, or foraging habitat loss. Collision is more likely to occur if seabirds fail to avoid wind farms, whereas displacement from foraging habitat is more likely to occur if seabirds do avoid wind farms. Thus displacement may be a consequence of avoidance behaviour. Disturbance, caused by structures or by human activity associated with wind farms, may also cause displacement if birds move away from wind farms due to associated disturbance. Here we define displacement as a reduced number of birds occurring within or immediately adjacent to offshore wind farms, and we define disturbance as birds spending extra time and/or energy to avoid structures or human activity associated with offshore wind farms. Birds may show two kinds of avoidance behaviour at offshore wind farms, often termed 'macro-avoidance' and 'micro-avoidance'. Macro-avoidance occurs when birds alter their flight path to keep clear of the whole wind farm (Desholm and Kahlert, 2005), whereas micro-avoidance occurs when birds enter the wind farm but take evasive action to avoid individual turbines (Band, 2011; Cook et al., 2012). If species-specific rates of macro-avoidance and micro-avoidance were known, it would be easy to assess vulnerability of different species' populations. However, data on macro-avoidance rates are very limited and inconsistent, while data on micro-avoidance by marine birds at offshore wind farms are extremely scarce (Cook et al., 2012). For example, it has been stated that northern gannets *Morus bassanus* show higher macro-avoidance (64% Krijgsveld et al., 2011) than shown by gulls. However, while Krijgsveld et al. (2011) report an 18% macro-avoidance rate by gulls, Petersen et al. (2006) report a 76% macro-avoidance rate by gulls, higher than that reported for gannets. Until avoidance rates are better quantified we have assumed that avoidance rates are not significantly different among taxa, since existing data fail to contradict that simplification.

The increase in electricity generation from offshore wind creates an urgent need to assess effects on marine birds, and potential impacts on their populations. The most detailed studies of the effects of wind farms on marine birds have been at the Nysted and Horns Rev offshore wind farms in Danish waters, where

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research has demonstrated a low risk of collision, though for a limited set of species, notably common eiders *Somateria mollissima* (Desholm and Kahlert, 2005). Fox et al. (2006a,b) however emphasised the high variability in behavioural responses and thus vulnerability amongst marine bird species to offshore wind farms. Due to this variability in behavioural responses, it is vital to identify species' populations most likely to be at risk. In Europe, many marine birds breed within Special Protection Areas (SPAs) and so are protected by European law, in particular the Birds Directive (2009/147/EC). This legal protection has the potential to constrain development of offshore wind farms so there is a strong incentive to improve understanding of the risks and to develop better directed monitoring. Offshore wind in the United States similarly needs to consider possible impacts on marine bird populations in the context of protection afforded by legislation such as the Endangered Species Act.

A traditional method in conservation for setting priorities is to develop lists of at-risk species (Gardali et al., 2012) but a means by which to identify which species are most vulnerable is therefore necessary. A well-established approach has been to use indices of sensitivity or population vulnerability to particular hazards (Furness and Tasker, 2000; Gardali et al., 2012; Garthe and Hüppop, 2004; Sonntag et al., 2012). Garthe and Hüppop (2004) developed an index of marine bird population vulnerability to offshore wind farms, based on scores of conservation importance of different species' populations and perceived behaviour-related risks of collision and displacement, combined into a single index. They applied this index to marine birds in the southern North Sea. Here we develop and refine the approach advocated by Garthe and Hüppop (2004), extending the species list and incorporating new data from recent research on the flight behaviour of marine birds. An important innovation in our approach is the separation of assessments of risk due to collision and risk due to avoidance/displacement. This separation is particularly useful where the species most at risk of collision differ from those most at risk of displacement, as is the case here. Our assessment focuses on the example of marine birds in Scotland, but the approach could equally be applied elsewhere.

2. Methods

We followed the approach of Garthe and Hüppop (2004) scoring broadly similar factors in the assessment of vulnerability of marine bird populations to offshore wind farms. Where we used the same factors as Garthe and Hüppop (2004), scores allocated by them were reviewed individually. In many cases those scores appeared appropriate but where more recent data were available, scores were modified. As far as possible, the scoring criteria for each factor and the respective provisional scores for each marine bird species were evidence-based, with data taken from the reviewed literature. Scoring criteria and provisional scores were subsequently circulated to a group of appropriate experts for review, in order to ensure consensus support for the final criteria and scorings from a range of stakeholders; including seabird ecologists and conservationists (see details in Acknowledgements). The list of marine bird species assessed includes true seabirds, wintering sea ducks and grebes, and white-tailed eagle *Haliaeetus albicilla*. This last species was included because it feeds extensively over the sea, and there are detailed studies of its collisions at coastal terrestrial wind farms. Both breeding and nonbreeding (wintering or passage) marine birds were scored. A few marine bird species of conservation importance that occur in Scotland were omitted, on the basis that they occur very infrequently, if ever, in Scottish marine areas identified as suitable for offshore wind farms (Marine Scotland, 2011). These included red-breasted merganser *Mergus serrator* and little gull *Hydrocoloeus minutus*.

The method considers factors representing conservation importance and factors representing aspects of sensitivity to wind farm impacts. In some regions, a single factor may be adequate to rank and assign scores for conservation importance, but often there are multiple rankings available, based on slightly different criteria. In such cases an average of several rankings may provide a better scoring system than relying on any one individual factor. In our case study, four factors represent the conservation importance of the species in a Scottish context; status in relation to the Birds Directive, percentage of the biogeographic population that occurs in Scotland, adult survival rate, and UK threat status. Status in relation to the Birds Directive gives emphasis to species considered by the European Commission to be in particular need of conservation protection through European legislation. However, some of these species show increasing populations. UK threat status emphasises species showing population decline, but may give a low score to populations of high endemism if these are not in decline. Percentage of biogeographic population in the focal area emphasises endemism but does not take account of population trend. Considering adult survival rate recognises that added mortality of adult birds with high natural survival rates (and corresponding low productivity) has a greater impact on population dynamics than added mortality to populations with low survival rates. Combining these factors appears to provide a better ranking of conservation importance than achieved by any component factor (for details of these factors and scores see Furness et al., 2012).

Six factors represent aspects of species' behaviour that contribute to their potential vulnerability to wind farms (flight altitude, flight manoeuvrability, percentage of time flying, nocturnal flight activity, disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation). Scores were assigned on a scale of 1–5 for almost all factors, where 5 is a strong anticipated negative impact. It was felt more appropriate to score the factor assessing flight altitude as a percentage value of a species' flight altitude spent at turbine blade height, rather than on a five point scale. Individual factor scores were combined to give a total for each species that ranked species according to their vulnerability to offshore wind farm developments separately in terms of collision risk and habitat loss through avoidance. The factors assessed and calculations used to combine factor scores are outlined below.

2.1. Flight altitude

This factor is widely considered to be of overwhelming importance in determining the risk of collision of marine birds with offshore wind turbines (Band, 2011; Cook et al., 2012). Birds that only fly very low over the water will be below the area swept by turbine blades, whereas marine birds that habitually fly at greater heights may experience a greater risk of collision with blades if flight heights coincide with rotor swept areas of a wind farm. Flight altitude scores were initially taken from Garthe and Hüppop (2004), and Cook et al. (2012) but we present flight altitude as percentages of birds flying at blade height (Table 1), as opposed to collapsing such data into a 5 point scale (Garthe and Hüppop, 2004). When there was a lack of consensus between Garthe and Hüppop (2004) and Cook et al. (2012), we estimated a flight altitude that was consistent with these two reviews and was consistent with data from closely related species, or with other published data such as Rothery et al. (2009), and Krijgsveld et al. (2011), trying to account for very different sample sizes in different studies, different methods of measurement, and trying to retain consistency across related species. Flight altitude includes birds in all activities (such as foraging, commuting, migrating). It may vary seasonally, but there are too few data available at present to test this possibility.

Table 1
Estimated percent of flight at turbine blade height (ca. 20–150 m asl).

Species	Reference	Estimated % at blade height
Greater scaup	Dirksen et al., 1998 (10%, no sample size quoted, radar at night). Assumed similar to other ducks.	3
Common eider	Krüger and Garthe, 2001 (2%, $n = 14{,}405$, land-based obs) Day et al., 2003 (0%, $n = 17$, land-based obs) Garthe and Hüppop, 2004 (score of 1, median height <5 m). Leopold et al., 2004 (1.3%, $n = 235$, ship-based obs) Petterson, 2005 (Utgrunden, 0%, $n = 57$, radar) Petterson, 2005 (Yttre Stengrund, 20%, $n = 2044$, radar) Sadoti et al., 2005a (0%, $n = 84$, boat-based obs) Petersen et al., 2006 (16%, $n = 193$ flocks mostly eiders, radar) Larsen and Guillemette, 2007 (2%, $n = 1277$ platform-based obs) Rothery et al., 2009 (0%, $n = 1282$, land-based obs) Paton et al., 2010 (0.2%, $n = 24{,}195$ land-based obs) Paton et al., 2010 (0%, $n = 294$ ship-based obs)	3
Long-tailed duck	Day et al., 2003 (0%, $n = 108$, land-based obs) Paton et al., 2010 (0%, $n = 280$ land- or ship-based obs) Cook et al., 2012 (0%, $n = 114$ ship-based obs) Our estimate assumes this species is similar to other ducks.	3
Common scoter	Krüger and Garthe, 2001 (3%, $n = 6754$, land-based obs) Garthe and Hüppop, 2004 (score of 1, median height <5 m) Leopold et al., 2004 (0.8%, $n = 2258$, ship-based obs) Sadoti et al., 2005a (43%, $n = 96$, boat-based obs) Npower renewables, 2006 (0.2%, $n = 1274$, boat-based obs) Rothery et al., 2009 (0%, $n = 341$, land-based obs) Paton et al., 2010 (0.2%, $n = 4756$ land-based obs) Paton et al., 2010 (0%, $n = 277$ ship-based obs) Cook et al., 2012 (1%, 95% ci <0.1–17%, $n = 30{,}847$ ship-based obs) Garthe et al., 2012b (24%, no sample size, boat-based obs)	3
Velvet scoter	Day et al., 2003 (0%, $n = 5$, land-based obs) Garthe and Hüppop, 2004 (score of 1, median height <5 m); Sadoti et al., 2005a (3%, $n = 88$, boat-based obs) Paton et al., 2010 (7.0%, $n = 2973$ land-based obs) Paton et al., 2010 (0%, $n = 161$ ship-based obs) Cook et al., 2012 (0%, $n = 20$ ship-based obs)	3
Common goldeneye	Dirksen et al., 1998 (5%, no sample size quoted, radar at night) Paton et al., 2010 (11.3%, $n = 336$ land-based obs) Assumed similar to other ducks	3
Red-throated diver	Krüger and Garthe, 2001 (0%, $n = 247$, land-based obs) Garthe and Hüppop, 2004 (score of 2, median height 5–10 m) Leopold et al., 2004 (8.5%, $n = 284$, ship-based obs, may include a few black-throated divers) Krijgsveld et al., 2005 (24%, $n = 103$, radar, may include a few black-throated divers) Sadoti et al., 2005a (3%, $n = 28$, boat-based obs) Npower renewables, 2006 (0%, $n = 13$, boat-based obs) Paton et al., 2010 (7.1%, $n = 1226$ land-based obs) Paton et al., 2010 (28.3%, $n = 106$ ship-based obs) Krijgsveld et al., 2011 (5%, no sample size, radar) Cook et al., 2012 (2%; 95% ci <0.1–22%, $n = 9715$ ship-based obs)	5
Black-throated diver	Garthe and Hüppop, 2004 (score of 2, median height 5–10 m); Cook et al., 2012 (0.1%; 95% ci <0.1–30%, $n = 126$ ship-based obs) Score is a compromise between conflicting data in these two studies and a view from reviewers that all divers should be same.	5
Great northern diver	Sadoti et al., 2005a (4%, $n = 27$, boat-based obs) Paton et al., 2010 (22.8%, $n = 2762$ landb-ased obs) Paton et al., 2010 (5.8%, $n = 292$ ship-based obs) Cook et al., 2012 (0%, $n = 14$ ship-based obs); Kerlinger, 1982 (migration can occur at 1000 to 3000 m heights)	5
Great-crested grebe	Garthe and Hüppop, 2004 (score of 2, median height 5–10 m); Leopold et al., 2004 (0%, $n = 32$, ship-based obs) Cook et al., 2012 (0%, $n = 82$ ship-based obs)	4
Slavonian grebe	Paton et al., 2010 (23.5%, $n = 85$ land-based obs) Assumed similar to great-crested grebe	4
Northern fulmar	Garthe and Hüppop, 2004 (score of 1, median height <5 m); Leopold et al., 2004 (0%, $n = 178$, ship-based obs) Krijgsveld et al., 2005 (0%, $n = 10$, radar) Cook et al., 2012 (0.2%; 95% ci <0.1–22%, $n = 29{,}168$ ship-based obs).	1
Sooty shearwater	Paton et al., 2010 (0%, $n = 5$ land-based obs) Paton et al., 2010 (0%, $n = 16$ ship-based obs) Cook et al., 2012 (0%, $n = 2$ ship-based obs) Assumed similar to Manx shearwater	0
Manx shearwater	Cook et al., 2012 (0.04%; 95% ci <0.01–10%, $n = 6957$ ship-based obs)	0
European storm-petrel	Cook et al., 2012 (2% (range 0–2.5%), $n = 52$ ship-based obs)	2
Leach's storm-petrel	Assumed similar to European storm-petrel	2

Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Northern gannet	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Leopold et al., 2004 (13%, n = 803, ship-based obs) Krijgsveld et al., 2005 (44%, n = 143, radar, excludes birds following fishing boats) Sadoti et al., 2005a (20%, n = 85, boat-based obs) Npower renewables, 2006 (16%, n = 50, boat-based obs) Rothery et al., 2009 (13%, n = 414, land-based obs) Paton et al., 2010 (10%, n = 8560 land-based obs) Paton et al., 2010 (6.9%, n = 1278 ship-based obs) Krijgsveld et al., 2011 (30%, no sample size, radar) Cook et al., 2012 (estimated 9.6%; 95% ci <0.1–20%, n = 44,851); Garthe et al., 2012b (9%, no sample size, boat-based obs)	16
Great cormorant	Garthe and Hüppop, 2004 (score of 1, median height <5 m). Leopold et al., 2004 (7.5%, n = 929, ship-based obs) Rothery et al., 2009 (13%, n = 352, land-based obs) Npower renewables, 2006 (7%, n = 113, boat-based obs) Paton et al., 2010 (7.3%, n = 2014 land-based obs) Paton et al., 2010 (0%, n = 15 ship-based obs) Krijgsveld et al., 2011 (28%, no sample size, radar)	4
Shag	Npower renewables, 2006 (0%, n = 5, boat-based obs) Cook et al., 2012 (12.4% with model fit relatively poor; 95% ci 1.9–60%, n = 233 ship-based obs).	5
White-tailed eagle	Nygård et al., 2010 (24% of flights in study wind farm were at blade height (hub height = 70 m, blade radius = 38–41 m)	24
Arctic skua	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Npower renewables, 2006 (0%, n = 2, boat-based obs) Paton et al., 2010 (21.1%, n = 19 land-based obs) Paton et al., 2010 (0%, n = 1 ship-based obs) Cook et al., 2012 (3.8%; 95% ci <0.1–16%, n = 331 ship-based obs) Observations of Arctic skuas from seawatching and from birds foraging at sea in breeding areas suggest higher flying than Cook et al., 2012 model, as does G&H 2004 score. Our estimate follows Garthe and Hüppop, 2004 and suggestions from reviewers more closely than the data in Cook et al. (2012).	10
Great skua	Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Cook et al., 2012 (4.3%; 95% ci 1.2–28%, n = 1202 ship-based obs) Observations of great skuas from seawatching, from birds foraging at sea in breeding areas, and from deployment of GPS data loggers by H. Wade, C. Thaxter and colleagues) suggest higher flying than Cook et al., 2012 model, as does G&H 2004 score. Our estimate follows Garthe and Hüppop, 2004, unpublished GPS logger data, and suggestions from reviewers more closely than data in Cook et al. (2012).	10
Black-headed gull	Bergh et al., 2002 (4%, n = 82, land-based obs at Slufterdam) Bergh et al., 2002 (78%, n = 41, land-based obs at Slag Dobbelsteen) Scored 5 by Garthe and Hüppop (2004) (median height 10–20 m with 10% above 100 m). Krijgsveld et al., 2005 (25%, n = 334, radar, excludes birds following fishing boats) Rothery et al., 2009 (4%, n = 978, land-based obs) Krijgsveld et al., 2011 (30%, no sample size, radar) Cook et al., 2012 (7.9%; 95% ci 0.4–50%, n = 4490 ship-based obs). Estimate also considers values for related gull species, and radar studies reporting a higher flight height than obtained from boat-based windfarm surveys (Cook et al., 2012)	18
Common gull	Bergh et al., 2002 (58%, n = 120, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 3, median height 10–20 m but few above 50 m); Krijgsveld et al., 2005 (48%, n = 1517, radar, excludes birds following fishing boats) Npower renewables, 2006 (18%, n = 102, boat-based obs) Cook et al., 2012 (22.9%; 95% ci 8.5–47%, n = 10,190 ship-based obs); Garthe et al., 2012b (11%, no sample size, boat-based obs)	23
Lesser black-backed gull	Bergh et al., 2002 (34%, n = 92, land-based obs at Slufterdam) Bergh et al., 2002 (90%, n = 1828, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 4, median height 10–20 m with 10% above 50 m); Krijgsveld et al., 2005 (55%, n = 2470, radar, excludes birds following fishing boats) Npower renewables, 2006 (35%, n = 66, boat-based obs) Krijgsveld et al., 2011 (60%, no sample size, radar) Cook et al., 2012 (estimated 25.2%; 95% ci 7.8–52%, n = 35,142 ship-based obs); Garthe et al., 2012b (29%, no sample size, boat-based obs)	30
Herring gull	Bergh et al., 2002 (33%, n = 71, land-based obs at Slufterdam) Bergh et al., 2002 (84%, n = 7327, land-based obs at Slag Dobbelsteen) Garthe and Hüppop, 2004 (score of 4, median height 10–20 m with 10% above 50 m); Krijgsveld et al., 2005 (50%, n = 2223, radar, excludes birds following fishing boats) Sadoti et al., 2005a (22%, n = 63, boat-based obs) Sadoti et al., 2005b (5%, n = 63, boat-based obs) Npower renewables, 2006 (37%, n = 142, boat-based obs) Rothery et al., 2009 (33%, n = 1408, land-based obs) Paton et al., 2010 (15.0%, n = 51,036 land-based obs) Paton et al., 2010 (13.8%, n = 1652 ship-based obs) Krijgsveld et al., 2011 (55%, no sample size, radar) Cook et al., 2012 (estimated 28.4%; 95% ci 16–48%, n = 25,252 ship-based obs); Garthe et al., 2012b (40%, no sample size, boat-based obs)	35

(*continued on next page*)

Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Great black-backed gull	Garthe and Hüppop, 2004 (Score 3, median height 10–20 m but few above 50 m) Krijgsveld et al., 2005 (57%, $n = 143$, radar, excludes birds following fishing boats) Sadoti et al., 2005a (28%, $n = 163$, boat-based obs) Sadoti et al., 2005b (6%, $n = 35$, boat-based obs) Npower renewables, 2006 (23%, $n = 22$, boat-based obs) Rothery et al., 2009 (44%, $n = 564$, land-based obs) Paton et al., 2010 (8.5%, $n = 8610$ land-based obs) Paton et al., 2010 (8.8%, $n = 1001$ ship-based obs) Krijgsveld et al., 2011 (60%, no sample size, radar) Cook et al., 2012 (33.1%; 95% ci 18–57%, $n = 8911$ ship-based obs) Garthe et al., 2012b (56%, no sample size, boat-based obs)	35
Black-legged kittiwake	Day et al., 2003 (10%, $n = 36$, land-based obs) Garthe and Hüppop, 2004 (Score 2, median 5–10 m) Leopold et al., 2004 (9%, $n = 637$, ship-based obs) Chamberlain et al., 2005 (4%, $n = 2036$, ship-based obs) Krijgsveld et al., 2005 (38%, $n = 2459$, radar, excludes birds following fishing boats) Npower renewables, 2006 (8%, $n = 79$, boat-based obs) Rothery et al., 2009 (11%, 1350, land-based obs) Paton et al., 2010 (0%, $n = 56$ land-based obs) Paton et al., 2010 (10.9%, $n = 55$ ship-based obs) Krijgsveld et al., 2011 (45%, no sample size, radar) Cook et al., 2012 (15.7%; 95% ci 8–24%, $n = 62{,}975$ ship-based obs); Garthe et al., 2012b (18%, no sample size, boat-based obs)	16
Little tern	Everaert and Stienen, 2007 (23%, $n = 2124$, land-based obs at a coastal wind farm) Assumed similar to other terns	7
Sandwich tern	Krüger and Garthe, 2001 (0%, $n = 959$, land-based obs) Garthe and Hüppop, 2004 (Score 3, median height 10–20 m but few above 50 m) Leopold et al., 2004 (4.5%, $n = 419$, ship-based obs) Krijgsveld et al., 2005 (56%, $n = 236$, radar, may include some other tern species) Npower renewables, 2006 (5%, $n = 20$, boat-based obs) Everaert and Stienen, 2007 (10%, $n = 27{,}571$, land-based obs at a coastal wind farm) Rothery et al., 2009 (3%, $n = 2137$, land-based obs) Perrow et al., 2011a (48%, no sample size, boat-based obs) Krijgsveld et al., 2011 (50%, no sample size, radar) Cook et al., 2012 (3.6%; 95% ci 0.7–35%, $n = 33{,}392$ ship-based obs); Garthe et al., 2012b (8%, no sample size, boat-based obs)	7
Common tern	Bergh et al., 2002 (8%, $n = 113$, land-based obs at Slufterdam) Krüger and Garthe, 2001 (0%, $n = 271$, land-based obs) Garthe and Hüppop, 2004 (score of 2, median 5–10 m) Leopold et al., 2004 (0.6%, $n = 1011$, ship-based obs, includes some Arctic terns) Sadoti et al., 2005a (6.2%, $n = 130$, boat-based obs 2003; 5.7%, $n = 163$, boat-based obs 2004, may include some roseate terns) Sadoti et al., 2005b (4%, $n = 29$, boat-based obs) Npower renewables, 2006 (41%, $n = 79$, boat-based obs) Everaert and Stienen, 2007 (13%, $n = 14{,}491$, land-based obs at a coastal wind farm) Hatch and Brault, 2007 (6.2%, $n = 3154$ (including some roseate terns) Paton et al., 2010 (3.8%, $n = 3644$ land-based obs) Paton et al., 2010 (11.5%, $n = 61$ ship-based obs) Cook et al., 2012 (12.7%; 95% ci 6–19%, $n = 19{,}332$ ship-based obs); Garthe et al., 2012b (0%, no sample size, may include a few Arctic terns)	7
Roseate tern	Paton et al., 2010 (0%, $n = 125$ land-based obs) Paton et al., 2010 (12.5%, $n = 8$ ship-based obs) Assumed similar to other terns	5
Arctic tern	Alerstam and Gudmundsson, 1999 (migrate mainly above collision risk height from 200 to 2000 m, radar) Gudmundsson et al., 2002 (migration mainly nocturnal, mean height 800 m, radar) Garthe and Hüppop, 2004 (score of 1, median <5 m); Cook et al., 2012 (2.8%; 95% ci <0.1–23%, $n = 2571$ ship-based obs)	5
Common guillemot	Day et al., 2003 (0%, $n = 172$, land-based obs, may include a few Brunnich's guillemots) Garthe and Hüppop, 2004 (score of 1, median <5 m); Leopold et al., 2004 (0.6%, $n = 316$, ship-based obs) Krijgsveld et al., 2005 (3%, $n = 440$, radar, may include some razorbills and puffins) Npower renewables, 2006 (5%, $n = 56$, boat-based obs) Paton et al., 2010 (0%, $n = 131$ ship-based obs) Cook et al., 2012 (0.01%; 95% ci <0.01–3.9%, $n = 36{,}260$ ship-based obs) In breeding areas, often commutes to/from colony at heights relating to altitude of nest sites which can be up to 200 m above sea level	1

Table 1 (*continued*)

Species	Reference	Estimated % at blade height
Razorbill	Garthe and Hüppop, 2004 (score of 1, median <5 m); Leopold et al., 2004 (0%, $n = 33$, ship-based obs) Sadoti et al., 2005a (0%, $n = 3$, boat-based obs) Npower renewables, 2006 (0%, $n = 17$, boat-based obs) Paton et al., 2010 (0%, $n = 135$ land-based obs) Paton et al., 2010 (0%, $n = 93$ ship-based obs) Cook et al., 2012 (0.4%; 95% ci <0.1–25%, $n = 13{,}171$ ship-based obs) In breeding areas, often commutes to/from colony at heights relating to altitude of nest sites which can be up to 200 m above sea level	1
Black guillemot	Paton et al., 2010 (0%, $n = 1$ land-based obs) Assume similar to closely related species	1
Little auk	Paton et al., 2010 (0%, $n = 125$ ship-based obs) Cook et al., 2012 (0.03%; 95% ci <0.01–15%, $n = 1287$ ship-based obs)	1
Atlantic puffin	Garthe and Hüppop, 2004 (score of 1, median <5 m); Paton et al., 2010 (0%, $n = 5$ ship-based obs) Cook et al., 2012 (0.1%; 95% ci <0.1–8%, $n = 5981$ ship-based obs)	1

2.2. Flight manoeuvrability

This factor takes into account the aerial agility of species and hence their potential to avoid collision with wind turbines at sea. Following Garthe and Hüppop (2004), we assume that, all other factors being equal, birds with low flight manoeuvrability are more likely to collide with wind turbines at offshore wind farms than birds with high flight manoeuvrability. Scores were taken from Garthe and Hüppop (2004), but adjusted where more recent data suggest appropriate. For additional species, scores were based on peer-reviewed literature and subjective judgement moderated by expert opinion. Species were classified from ‘very high flight manoeuvrability’ (score 1) to ‘very low manoeuvrability’ (score 5) (electronic Appendix 1). The value is considered to be a consequence of morphology rather than behaviour. It may vary seasonally (for example in relation to moult) but such detail is beyond the scope of this assessment.

2.3. Percentage of time flying

This factor was assumed to indicate risk of collision because marine birds that spend more time flying while at sea (whether while breeding, migrating, wintering, or as prebreeders) are more likely to be at risk of collision. Where available, scores were taken from Garthe and Hüppop (2004) and adjusted where more recent data suggest appropriate. For other species, scores were calculated from data on activity budgets following the procedure outlined by Garthe and Hüppop (2004). Species were scored 1 if 0–20% of time at sea was spent in flight, 2 if 21–40% was spent flying, 3 if 41–60% was spent flying, 4 if 61–80% was spent flying, and 5 if 81–100% was spent flying (electronic Appendix 2). This factor will probably vary seasonally, with the literature indicating more flight activity while rearing chicks than during the incubation period, and more flight while breeding than during winter. Peaks of flight activity occur in migrant species during the migration, while flight activity may be reduced during post-breeding moult. However, these refinements are not yet well enough documented to assess scores separately for different seasons, although that could be a useful development of the method.

2.4. Nocturnal flight activity

Although various observations exist, detailed data on nocturnal flight activity are not available for many species. Geolocation and GPS logger data are starting to change this situation, predominantly for large Southern Ocean seabirds (Mackley et al., 2010, 2011; Phalan et al., 2007), but with data now available for northern gannet (Garthe et al., 2012a) and black-legged kittiwake *Rissa tridactyla* (Kotzerka et al., 2010). Similar data should soon be available for some other North Atlantic seabirds including great skuas *Stercorarius skua* for which logger deployments have already provided data on migrations and wintering areas (Magnusdottir et al., 2012). We used scores published in Garthe and Hüppop (2004) for the species where these were available: Score 1 (limited flight activity at night) to score 5 (much flight activity at night) (electronic Appendix 3). We used published data where possible, and information (often qualitative rather than quantitative) from individual species studies or from handbooks (Cramp and Simmons, 1977, 1980; del Hoyo et al., 1992, 1996; Glutz von Blotzheim and Bauer, 1982). Classifications were also moderated by experts.

2.5. Disturbance by wind farm structures, ship and helicopter traffic

Marine bird species vary in their reactions to offshore wind farms and the ship and helicopter traffic that occurs during maintenance of the turbines. Where available, scores were taken from Garthe and Hüppop (2004) and adjusted where more recent data were available (e.g. Schwemmer et al., 2011). For additional species, information on disturbance sensitivity was taken from the peer-reviewed literature, and scores were moderated by experts. Scoring categories were: 1 (limited escape behaviour and a very short flight distance when approached), to 5 (strong escape behaviour, at a large response distance) (electronic Appendix 4).

2.6. Habitat specialisation

Marine birds vary in the range of habitats they use, for example relating to water masses and frontal systems and whether they use these as specialists or generalists. This score classifies species into categories from 1 (tend to forage over large marine areas with little known association with particular marine features) to 5 (tend to feed on very specific habitat features, such as shallow banks with bivalve communities, or kelp beds) (electronic Appendix 5). Where available, scores presented by Garthe and Hüppop (2004) were used. Scores for other species were based on foraging ecology described in single species studies in the literature, or from standard handbook descriptions.

2.7. Calculating species vulnerability scores for collision risk and displacement concern

Garthe and Hüppop (2004) calculated species vulnerability scores according to Equation (1), where *e*, *f*, *g*, *h*, *i* and *j* represent flight altitude, flight manoeuvrability, percentage of time flying, nocturnal flight activity, disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively.

$$\text{Species vulnerability score} = (e+f+g+h)/4 \times (i+j)/2 \times \text{conservation score} \quad (1)$$

This recognised that the first four factors all relate to flight ability and flight behaviour, while the last two factors relate to habitat use and susceptibility to disturbance. Thus their index combined both collision risk and disturbance/habitat loss considerations into a single score.

We use an alternative approach and score separately for collision concern and for disturbance/habitat displacement concern. For collision risk, we give a high weighting to flight altitude (*e*), and lower weighting to manoeuvrability (*f*), percentage of time flying (*g*), and nocturnal flight activity (*h*) (Equation (2)).

$$\text{Collision risk score} = e \times (f+g+h)/3 \times \text{conservation importance score} \quad (2)$$

For disturbance/habitat displacement we calculated a vulnerability index according to Equation (3) where *i* and *j* represent disturbance by wind farm structures, ship and helicopter traffic, and habitat specialisation respectively. We divided the total by 10 (an arbitrary value) to recognise that the disturbance/displacement impact on populations is likely to be considerably less than a direct mortality impact such as from collisions and therefore the two scales should not be compared in a quantitative way but only in terms of the species ranking within one scale.

$$\text{Disturbance/displacement score} = ((\mathrm{i} \times \mathrm{j}) \times \text{conservation importance score})/10 \quad (3)$$

3. Results

Factors and species-scores were sent to a panel of seabird experts for comment. Most reviewers suggested no change to the factors used, and no change to most scores. Most reviewers felt that one or two out of the 228 scores should be adjusted, so agreed with more than 99% of the scores. The few scores that were consistently questioned by reviewers were altered to bring them in line with this consensus opinion.

The ranked species' vulnerability scores in the context of collision risk (and considering the conservation importance of the species) are summarised in Table 2. The percentage of a species' flight altitude at turbine blade height ranged from values of 0–35. The mean of the other factors ranged from 1.3 to 3.7 (within a theoretically possible range of 1–5). Details of the derivation of the component scores are given in Table 2 and the electronic Appendix. The highest scores indicate that the larger gull species (great black-backed gull *Larus marinus*, herring gull *Larus argentatus*, and lesser black-backed gull *Larus fuscus*), northern gannet and white-tailed eagle are the species likely to be most at risk of collision with offshore wind turbines.

The ranked species vulnerability scores in the context of disturbance or habitat displacement (and considering the conservation importance of the species) are summarised in Table 3. The highest scores indicate that all diver species and common scoter are most likely to be at risk of disturbance or displaced from habitats as a result of offshore wind turbines. Details of the derivation of the component scores are given in the electronic Appendix tables.

4. Discussion

The key results from our assessment are ranked species lists referring to species' vulnerability to offshore wind farms in the context of collision risk (Table 2) and vulnerability in the context of disturbance or habitat displacement (Table 3). For collision risk, the five top ranking species had scores that were greater than half of the maximum score achieved overall (by herring gull, with a collision risk score of 1306). These species were herring gull, great black-backed gull, lesser black-backed gull, white-tailed eagle and northern gannet. In the rankings for disturbance or habitat displacement concern, only three species had scores that were over half of the maximum score possible (maximum score possible was 50). In contrast to species indicated as having a high collision risk, the species identified as most vulnerable to disturbance or displacement were diver species and common scoter. Garthe and Hüppop (2004) identified divers and common scoter as the species with the highest sensitivity index scores in the southern North Sea, suggesting that their index weighted disturbance/displacement impacts more strongly than collision mortality risk. The differing results for the two potential risks shown by our assessment are a result of species-specific variability in behaviour. The species identified as of highest concern in the table rankings should be the focus for monitoring and for further research investigating the effects of offshore wind farms on marine birds in Scottish waters, and we suggest that this approach should be extended to consider marine bird communities in other countries.

4.1. Development/evolution of our approach

Following discussions and comments from reviewers, we developed the approach taken by Garthe and Hüppop (2004) to recognise that there is broad support for the view that collision concern should be considered separately from displacement concern, as the rankings of species in each context are very different. We also differ in our approach to Garthe and Hüppop (2004) by not ranking all scores for factors measuring the vulnerability of species to offshore wind turbines on a scale of 1–5. Instead, the factor assessing flight altitude was considered in greater detail by presenting the percentage of a species flight altitude spent at wind turbine blade height. In combination with an increased weighting given to this factor when calculating collision risk scores, our method incorporates the broad consensus of opinion among reviewers that flight height is considerably more important than the other factors considered here in assessing collision risk of marine birds with wind turbines. There were mixed views regarding the relative importance of the other three factors of manoeuvrability, percentage of time flying and the amount of nocturnal flight in affecting collision risk, and the down-weighting of these three factors recognised this. This change in approach is more appropriate now than it was when Garthe and Hüppop (2004) prepared their paper, because there are now considerably more detailed data on marine bird flight height from the work of Paton et al. (2010), Krijgsveld et al. (2011), Cook et al. (2012), Garthe et al. (2012b) and others.

4.2. Limitations of indices of sensitivity

The method of calculating indices and ranking species has been criticised for the uncertainty in how to incorporate a variety of different factors, often scored on different scales and sometimes with

Table 2
Ranked species concern in the context of collision impacts: percent flying at blade height × 1/3(manoeuverability score + % time flying score + nocturnal flight score) × conservation importance score (ranked by index value).

Species	Flight % at blade height	Flight agility	% of time flying	Night flight	Conservation importance score	Total risk score
Herring gull	35	2	2	3	16	1306
Great black-backed gull	35	2	2	3	15	1225
Lesser black-backed gull	30	1	2	3	16	960
White-tailed eagle	24	3	5	1	12	864
Northern gannet	16	3	3	2	17	725
Common gull	23	1	2	3	13	598
Black-legged kittiwake	16	1	3	3	14	523
Arctic skua	10	1	5	1	14	327
Great skua	10	1	4	1	16	320
Black-headed gull	18	1	1	2	12	288
Sandwich tern	7	1	5	1	15	245
Black-throated diver	5	5	3	1	16	240
Great northern diver	5	5	2	1	18	240
Common tern	7	1	5	1	14	229
Red-throated diver	5	5	2	1	16	213
Little tern	7	1	5	1	13	212
Arctic tern	5	1	5	1	17	198
Roseate tern	5	1	5	1	15	175
Shag	5	3	2	1	15	150
Slavonian grebe	4	4	2	2	13	139
Greater scaup	3	4	2	5	11	121
Common eider	3	4	2	3	13	117
Great cormorant	4	4	2	1	11	103
Common goldeneye	3	3	2	3	12	96
Common scoter	3	3	2	3	12	96
European storm-petrel	2	1	3	4	17	91
Velvet scoter	3	3	2	3	11	88
Leach's storm-petrel	2	1	3	4	16	85
Great-crested grebe	4	4	3	2	7	84
Long-tailed duck	3	3	2	3	8	64
Northern fulmar	1	3	2	4	16	48
Common guillemot	1	4	1	2	16	37
Razorbill	1	4	1	1	16	32
Black guillemot	1	4	1	1	13	30
Atlantic puffin	1	3	1	1	16	27
Little auk	1	3	1	1	9	15
Manx shearwater	0	3	3	3	17	0
Sooty shearwater	0	3	3	3	12	0

correlations among the factors being used (Desholm, 2009). Considering birds of all species and not just marine birds, Desholm (2009) argued that in order to prioritise bird species for assessment of the impact of collision mortality at offshore wind farms, it is possible to consider just two criteria; proportion of the biogeographic population at risk at a wind farm site, and population 'elasticity', which is mainly determined by adult survival rate. Birds with high adult survival rates (such as most marine birds) will be more severely impacted by wind farm mortality than birds with low natural survival rates (such as most terrestrial migrant passerines). That approach has the benefit of great simplicity, and works well when ranking bird species ranging from small passerines with very short life expectancy and high reproductive output through to marine birds with long life spans and low fecundity (Desholm, 2009). However, that method does not discriminate well between species with similar demographic parameters (most marine birds have similarly high adult survival rates), and it does not take into account the fact that some kinds of marine birds are more, or less, likely to collide with wind turbines as a consequence of their species-specific ecology or behaviour. For example, among marine birds with comparable demography, species that regularly fly at turbine blade height are apparently more at risk from collision mortality than species that only fly low over the water. Nor does that method take account of the possible effects of offshore wind farms on marine birds through effects such as deflection of flight paths (Masden et al., 2009; Speakman et al., 2009) or loss of foraging habitat (Fox et al., 2006a,b). In our analysis we are dealing mainly with species at the long-lived end of this spectrum of life histories and so the simple model focusing only on adult survival rate, but not on differences in ecology and behaviour, becomes less useful as there is relatively little variation in this among most marine bird species.

4.3. *Limitations of data sources, implications for results, and future improvements*

There are limitations inherent in the index resulting from data sources, which should be considered when interpreting our results. Flight altitude was considered to be the most important factor in calculating collision risk for marine bird species at offshore wind farms. In calculating the collision risk for species, Cook et al. (2012) provide the most detailed review of available data on flight altitude, however there are inherent limitations in these data, which should be considered. These include the fact that most data are from boat-based observations and as such, observations potentially include significant numbers of marine birds scared into flight by the boat (Camphuysen et al., 2004). Such birds tend to fly low over the water so will bias the distribution of flight heights. Birds of some species may also be attracted to boats. Additionally, most estimates of the height of flying marine birds used by Cook et al. (2012) are very crude, mostly being based on estimates by observers and not on measurements. It is worth noting that the data reported by Cook et al. (2012) can differ quite considerably to flight heights of

Table 3
Ranked species concern in the context of disturbance and/or displacement from habitat (Disturbance score × Habitat flexibility score × Conservation Importance score)/10.

Species	Disturbance by ship and helicopter traffic	Habitat use flexibility	Conservation importance score	Species concern index value
Black-throated diver	5	4	16	32
Red-throated diver	5	4	16	32
Great northern diver	5	3	18	27
Common scoter	5	4	12	24
Common goldeneye	4	4	12	19
Greater scaup	4	4	11	18
Velvet scoter	5	3	11	16
Common eider	3	4	13	16
Black guillemot	3	4	13	16
Slavonian grebe	3	4	13	16
Common guillemot	3	3	16	14
Razorbill	3	3	16	14
Shag	3	3	15	14
Great cormorant	4	3	11	13
Little tern	2	4	13	10
Arctic tern	2	3	17	10
Atlantic puffin	2	3	16	10
Long-tailed duck	3	4	8	10
Roseate tern	2	3	15	9
Sandwich tern	2	3	15	9
Common tern	2	3	14	8
Great-crested grebe	3	4	7	8
Great black-backed gull	2	2	15	6
Black-legged kittiwake	2	2	14	6
Common gull	2	2	13	5
Black-headed gull	2	2	12	5
Little auk	2	2	9	4
Northern gannet	2	1	17	3
Herring gull	2	1	16	3
Great skua	1	2	16	3
Lesser black-backed gull	2	1	16	3
Arctic skua	1	2	14	3
White-tailed eagle	1	2	12	2
Manx shearwater	1	1	17	2
European storm-petrel	1	1	17	2
Leach's storm-petrel	1	1	16	2
Northern fulmar	1	1	16	2
Sooty shearwater	1	1	12	1

marine birds that have been measured by radar (Krijgsveld et al., 2011; Cook et al., 2012). This discrepancy is unexplained, but it seems likely that the radar measurements of flight height are more reliable where confident of species attribution. Our index indicates that many marine bird species rarely fly at turbine blade height, and so appear to have negligible risk of population-level impacts from collision mortality. These include sea ducks, alcids, storm-petrels and shearwaters (though the possibility that such birds may occasionally fly higher than normal as a result of disturbance needs to be borne in mind). The low risk for these species is consistent with empirical data from long-established offshore wind farms (ICES, 2011). However, it would be desirable to have more data on flight heights to allow this inference to be converted into a confident conclusion that might permit species to be scoped out of assessments. In light of new, more detailed data it will be possible to revise the scorings presented here and establish with more confidence those species at risk. We do suggest however, that species presented here with high scores should be of particular concern in relation to offshore wind developments. Gulls, white-tailed eagles, northern gannets, skuas and divers are identified as being the groups whose populations are most at risk in a Scottish context.

The recent increase in the use of data loggers on seabirds is starting to provide more detailed information on the at-sea activity of seabirds. This should help achieve an increase in the amount of data available and improve the quality of flight altitude data and information required for some of the other factors considered in calculating collision risk (e.g. percentage of time spent in flight and nocturnal flight activity). It is also possible that, in the near future, collection of quantitative data on time spent in flight from geolocation data loggers (for example, based on salt-water switch recording time spent with the logger immersed in seawater) will allow scorings to be converted into a quantitative scale rather than the present qualitative one.

Previous studies indicate that some species of marine bird avoid wind farms and as such, collision levels are low (Fox et al., 2006b; ICES, 2011; Petterson, 2005; Lindeboom et al., 2011). Our collision risk index is likely to require modification in future as more data become available, and at present should be considered precautionary in that the relative avoidance responses of different species are not yet well known. In assessing the potential importance of displacement for different marine bird species, although there was strong consensus among reviewers for the scores used, this consensus may be more a result of uncertainty than confident agreement, and so the ranking of species needs to be treated with caution. However, we suggest that species with scores over 15 (divers, scoters, goldeneye *Bucephala clangula*, scaup *Aythya marila*, common eider, black guillemot *Cepphus grylle*, Slavonian grebe *Podiceps auritus*) should be considered as focal species for concern with regards to potential displacement effects. Species with scores below 8 (northern fulmar *Fulmarus glacialis*, storm-petrels, shearwaters, gulls, skuas, northern gannet, little auk *Alle alle*, and white-tailed eagle) seem very unlikely to be affected by displacement. It is worth noting that whilst it is clear that some marine birds do strongly avoid wind turbines at sea, recent work modelling the cumulative impact of disturbance by wind turbines suggests that the impact of these through increased travel distances and habitat loss is trivial, even for species that show especially strong avoidance behaviour and have a high displacement ranking, such as red-throated divers *Gavia stellata* (Topping and Petersen, 2011).

We are aware that our vulnerability index deals with a limited set of factors and that there are other potential impacts that are not necessarily covered. For example, Perrow et al. (2011b) presented evidence suggesting that little tern *Sternula albifrons* breeding success in a colony in Norfolk may have been reduced by a shortage of young herring *Clupea harengus* around Scroby Sands offshore wind farm caused by monopile installation affecting fish reproduction locally. To an extent, the high sensitivity of little tern would be indicated by our six factors because they are seabirds with a very short foraging range that utilise a very particular and restricted foraging habitat, so score as sensitive on the habitat flexibility factor. However, complex and indirect ecosystem effects such as alteration of fish and benthic invertebrate abundance by wind farms is something that is extremely difficult to predict, so caution is needed in interpretations and collecting post-construction monitoring data will be important. Once operational, offshore wind farms may possibly enhance food supplies for marine birds by acting as marine protected areas (e.g. closed to trawl fishing; Defew et al., 2012). Siting of wind farms can also be influential at specific sites. For example, turbines placed between a common tern *Sterna hirundo* colony and their feeding habitat have had a high impact on a particular colony (Everaert and Stienen, 2007; Stienen et al., 2008), which might not be the case where a wind farm is placed away from the obligatory flight line of birds from a specific breeding site. In addition, the perception of risk, which seems to vary among species, and possibly may be as important a factor as anatomical constraints, may also be relevant, but cannot readily be scored on a scale, though avoidance may possibly be a proxy for perception of risk. We considered these indirect and uncertain effects to be beyond the scope of this review, but emphasise that they should not be assumed to be negligible.

An additional factor that we could not easily consider here but should not be ignored is the possibility that weather conditions may affect collision risk for marine birds. For example, weather conditions such as fog or heavy rain, may obscure turbines. Such effects might over-ride any species-specific differences in vulnerability. In reviewing the results of studies at demonstration offshore wind farms in Denmark, Fox et al. (2006b) stated "observations at Nysted that waterbirds tend not to fly in the area of the turbines at night, or under adverse weather conditions (as found elsewhere; Petterson, 2005) suggest that collision risk is not likely to be high even under conditions when the turbines are less visible". These observations suggest that catastrophic mortality incidents caused by adverse weather conditions are less likely at offshore wind farms than has been suggested by some, although breeding birds may not have the flexibility to respond in this way. In addition, presence of lights on wind farms may affect collision risk for marine birds, but the susceptibility of different species to attraction to lights at sea is not well known (Merkel and Johansen, 2011).

4.4. Applications and future testing

In scoping potential areas for offshore wind farm development in Scottish waters, Davies and Watret (2011) considered constraints implied by seabird SPAs, and the distribution at sea of seabirds as indicated by the European Seabirds at Sea database. These data were combined with the flight height data presented by Cook et al. (2012) to assess numbers of marine birds flying at collision height risk in different parts of the Scottish marine area. The development of sensitivity scoring and conservation importance scoring for individual species of marine birds may help to refine such assessment of environmental constraints by allowing a focus on the marine bird species of greatest concern. This would most usefully be combined with mapping (e.g. Garthe and Hüppop, 2004) of the distribution of seabird SPAs and the numbers of each species protected at these sites.

The scores presented in this paper, and similar scoring for other species, will help developers preparing Environmental Statements for new sites, by providing a clearer focus on the key species likely to be of concern in relation to collision risk and in relation to displacement impacts. Nevertheless, these scores should be seen as iterative, requiring to be updated as more data become available, and possibly to be made obsolete when macro- and micro-avoidance rates are measured for a range of species at a range of offshore wind farms. The ranking of marine bird species in our collision risk index accords well with the concerns of offshore wind farm developers in Scottish waters, gulls and gannets being a particular concern at many offshore wind farms as expressed in Environmental Statements and Environmental Impact Assessments. However, there is a need for post-construction monitoring data from offshore wind farms to assess more fully the accuracy of these predictive indices, and there is an urgent need to study both macro-avoidance and micro-avoidance behaviour of different seabird species.

5. Conclusions

This paper advances previous work on likely impacts of offshore wind farms on marine bird populations by separating risks of collision mortality from risks of disturbance/displacement. These two aspects affect different marine bird species. An offshore wind farm collision risk index identifies populations of gulls, white-tailed eagles, northern gannets and skuas as the marine birds in Scottish waters most vulnerable to collision mortality impacts. In contrast, populations of divers and common scoters appear most vulnerable to displacement impacts. The collision risk index is particularly affected by the proportion of birds flying at collision risk height, and there is a need to collect more data on marine bird flight heights. Populations identified as most vulnerable should be the focus of future monitoring studies, but there is a need to develop research into macro- and micro-avoidance rates and how these vary among species.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2013.01.025.

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